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HABITAT TREE REQUIREMENTS AND THE EFFECTS OF REMOVAL DURING LOGGING ON THE MARSUPIAL BRUSH-TAILED PHASCOGALE (*PHASCOGALE TAPOATAFA TAPOATAFA*) IN WESTERN AUSTRALIA

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ABSTRACT

The characteristics of habitat (nesting) trees, their spatial organisation, and frequency of use by Brush-tailed Phascogales (*Phascogale tapoatafa tapoatafa*) were documented over a 3 year period in Jarrah forest of Western Australia. The level and direct effect of removal of habitat trees during logging operations was examined. 139 phascogales were radiotracked and 689 habitat trees were identified. 95% of these were either Marri (52.5%) or Jarrah (41.2%). Preference for use between these two tree species was unclear, due to unquantified environmental variability. Trees became markedly usable as habitat trees at above 40 cm dbhob (diameter at breast height over bark) and mean tree sizes used were 87 cm dbhob (Marri) and 76cm dbhob (Jarrah). Reuse rates of identified habitat trees showed that there was a significant difference ($p=0.05$) between the mean size of trees used more than once compared to un-reused trees, and larger trees, particularly those ≥ 95cm dbhob, represent higher quality habitat trees. Possums were also observed nesting in large or dead trees used by phascogales but measures of hollow entrances showed different entrance sizes were used and suggests that hollow competition is unlikely. Logging resulted in an almost total removal of all potential habitat trees from logging coupes containing phascogales, and phascogales completely ceased nesting in those parts of their territories affected by logging. They continued to forage extensively throughout coupes which had been cut during logging, but confined nesting to trees within unlogged corridors between cut areas. Post-logging surveys of four logging coupes revealed that potential habitat trees were not marked for retention at the policy rate of 3/hectare and of those marked few were suitable. Without change to current logging practices, it can be expected that phascogales will become locally extinct in areas extensively impacted by logging. Changes and recommendations to habitat tree retention policy in WA are discussed.

INTRODUCTION

The importance of habitat trees for hollow nesting species is well recognised (Recher *et al.* 1980, Lunney 1991, Lindenmayer 1994), but meaningful incorporation of this into forest management has been limited despite concerns and research into this topic since the 1970s (e.g. Tyndale-Biscoe & Calaby 1975). The retention of hollow-trees outside reserve areas and within regions affected by timber harvesting is now acknowledged as essential for the preservation of hollow-nesting fauna (Resource Assessment Commission (RAC) 1992).

In order to devise effective practices to retain habitat trees in managed forests, the central fauna issues have been to determine: what constitutes a habitat tree for any given species, how many of these are required per unit area, what is the required spatial configuration of such trees, and whether potential habitat trees and corridors retained after logging operations will actually be used by wildlife (e.g. Dickman 1991, Lindenmayer 1994).

The objectives of this study were to examine the above issues by collecting data on the nesting behaviour and habitat tree requirements of the hollow-dependent marsupial Brush-tailed Phascogale (*Phascogale tapoatafa tapoatafa*) in the Western Australian Jarrah (*Eucalyptus marginata*) forest. To investigate the effect of current logging practices on habitat tree availability, phascogales were additionally monitored before, during and after logging operations and surveys of retained habitat trees in cut areas were undertaken to quantify habitat tree availability after logging. Incidental information on possums was also gathered during the research. The study findings are presented here, with discussion on habitat tree retention requirements and policies in timber production areas of the Jarrah forest.

METHODS

The Study Species

The Brush-tailed Phascogale is an arboreal carnivorous marsupial which occurs throughout the dry sclerophyll forests of Australia. It is the largest member (typical mature females weights (W.A.): 93 – 140g, males 110 – 210g, Rhind, unpubl. data) of the dasyurid family that exhibit semelparity – breeding once in a lifetime – and all males die at the end of the annual winter mating season (Cuttle 1982). Females can have up to eight young, which take five months to raise, and may live to breed in a second year (a single individual from this study lived for three years, raising a third litter). The species is nocturnal and an obligate arborealist, dependent on trees for nest hollows and its main diet of bark-associated invertebrates. Males and females both have large territories of ≥20 hectares and female territories are

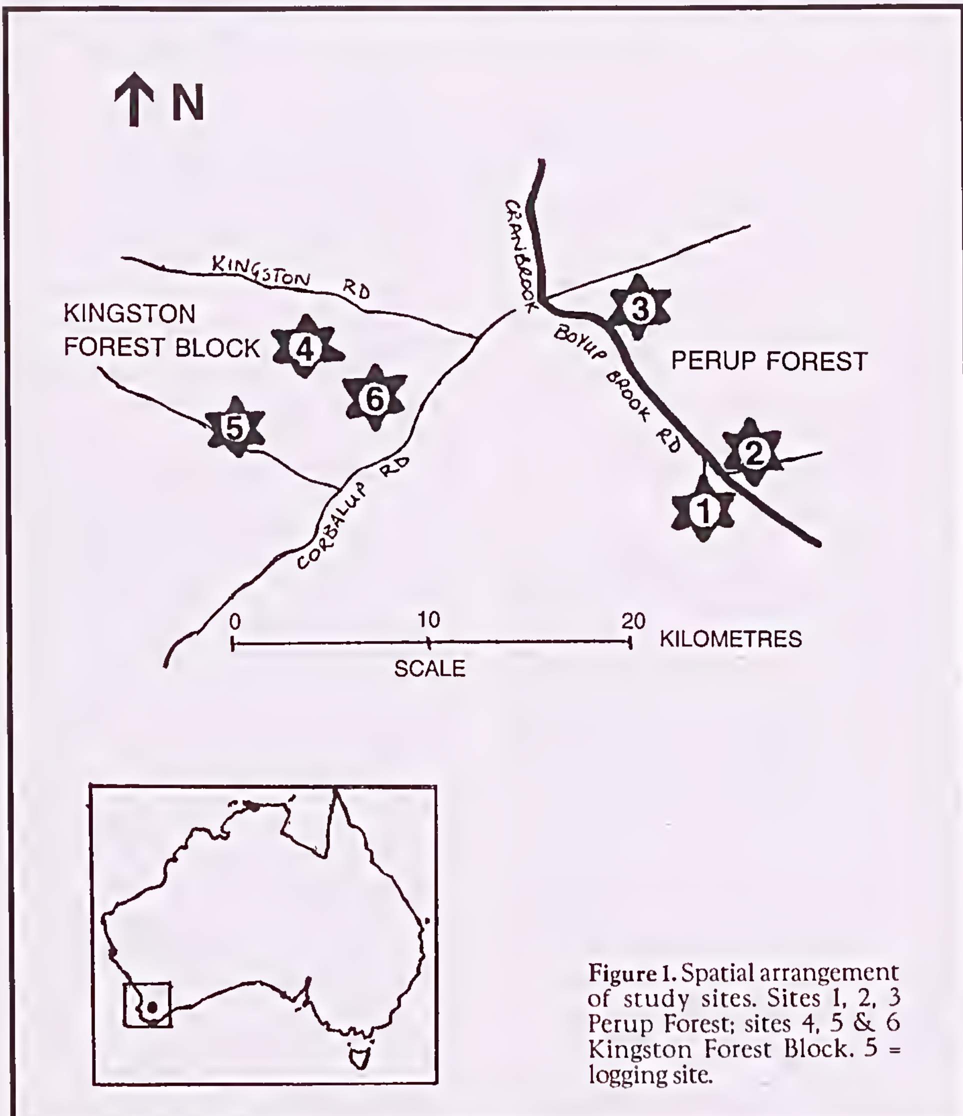

Figure 1. Spatial arrangement of study sites. Sites 1, 2, 3 Perup Forest; sites 4, 5 & 6 Kingston Forest Block. 5 = logging site.

exclusive, therefore animal densities are low. In Victorian populations, densities of females were estimated to be one individual per 50 hectares (Soderquist 1995a).

This species is generally regarded as rare Australia-wide, in that it is rarely seen or encountered during fauna surveying (Traill & Coates 1993) and its current status is Near Threatened (IUCN in press). In some areas it is genuinely rare. Phascogales are now considered extinct in South Australia and have disappeared from at least 40% of their previous range. The species is locally uncommon in parts of all States and its status is unknown for most of its remaining range. Details of the biology and behaviour of Brush-tailed Phascogales are discussed by Cuttle (1982), Traill & Coates (1993), Soderquist (1993a, 1994, 1995a, 1995b), Soderquist & Ealey (1994), and Soderquist & Lill (1995).

Study Sites

Study sites were established in Perup Forest (lat. 34°10' long. 116°35') (three sites) and 20 kilometres away in Kingston Forest Block (three sites) near Manjimup, Western Australia (Figure 1). Two Perup sites were established in 1992, a third Perup site was only used for three months (March–May 1994) during research on the effect of fire, and the three Kingston sites were established in 1993. Each site comprised a capture region of approximately 200 hectares, however study animals ranged well outside of these areas.

These regions receive an average annual rainfall of 600mm – 800mm (Bureau of Meteorology, Western Australia) and the forest is considered to be low quality (20–30m) mixed Jarrah and Marri (*E. calophylla*). The upper storey was dominated by these species, with small areas of Wandoo (*E. wandoo*), and Flooded Gum (*E. rudis*) and the understorey was generally open and comprised sparse scrub species. Details of the Jarrah forest are provided in Dell *et al.* (1989) and the study area is well described in "The Perup: A Living Forest" (CALM, undated).

Capture and General Study Approach

Wild Brush-tailed Phascogales were captured by retrieval from nestboxes and trapping. Trapping techniques, nestbox design, precautions involved in trapping for this species and the logistics of capturing phascogales are described by Traill & Coates (1993) and Soderquist & Ealey (1994). In this study, trapping intervals were set at 100 m along perimeter roads and along five 1.2–1.5km transects at each study site. Routine trapping was conducted several times each year between December and July, but was abandoned in 1995 due to escalating trap interference by other mammal species. In addition to trapping, permanent nestboxes were erected 3–4 m high on trees on a 300 X 300m interval grid throughout five of the sites (mean number of usable boxes per site over the course of the research = 26). Routine checking of nestboxes was conducted throughout each year and showed that all boxes were used by phascogales and not used by any other species.

Three hundred and sixty seven individual phascogales were captured during the study and were sexed, measured and individually

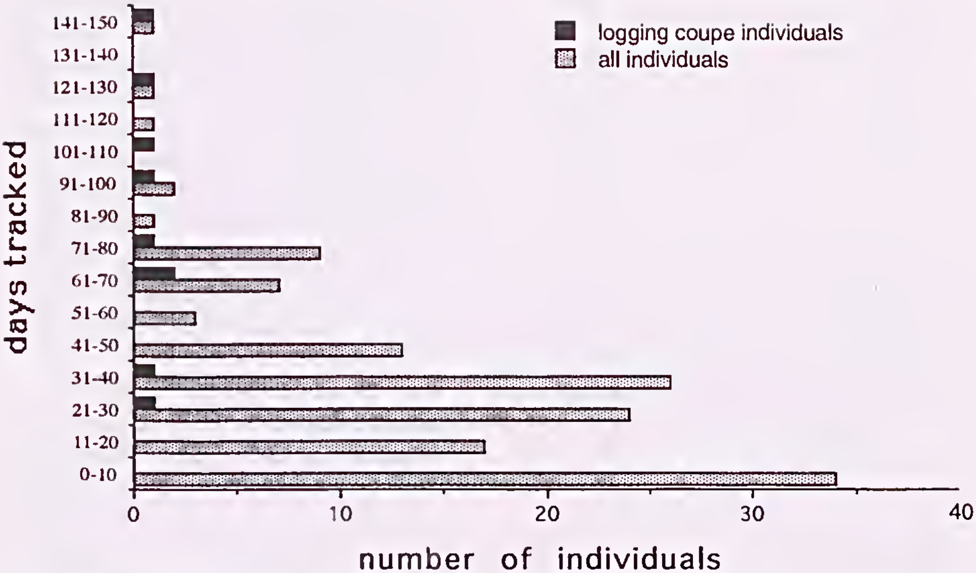

Figure 2. Number of days that radiocollared Brush-tailed Phascogales were tracked to habitat trees (phascogales n = 139, days tracked n = 4358).

marked by ear tattooing. One hundred and thirty nine of these were radiocollared using the 3–4g expanding break-away collar design of Soderquist (1993b). As part of a general study on the species, daily radio-tracking of phascogales was undertaken to identify nest-sites and determine nest site fidelity (frequency of use of any particular nest-site). Four thousand three hundred and fifty eight day/nest-site locations were recorded and 689 different trees were identified as nesting trees. Difficulty in identifying the exact tree being used by a collared animal occurred 5–10% of the time, varying with transmitter quality, topography and forest structure, and where there was uncertainty such trees were not recorded. Due to the large territories and long distance movements of phascogales (Soderquist 1995) the total area over which habitat trees were documented was estimated to encompass 2500–3500 hectares.

Individuals were tracked almost daily for up to 140 days, although 30–40 consecutive days was most typical. Figure 2 illustrates day-time radio-tracking effort per individual. The high incidence of tracking animals for <10 days was a consequence of: (i) collar loss due to collars being chewed off by the mothers of collared young and collars coming adrift due to insufficient securing at the tie point (ii) long distance (eg. 6 kilometres) dispersal of young from the natal nest (individuals could not be relocated) and (iii) predation of mothers and young at the vulnerable stages of weaning and dispersal.

In addition to day-time radio-tracking to habitat trees, night-time tracking and behavioural observations of active phascogales were undertaken to determine foraging and other behaviours, and to identify territory size and usage. Phascogales are typically undisturbed by observation using low illumination (Traill & Coates 1993), and could be quietly followed within metres. Behavioural observations and radio-tracking to determine territories were carried out using the methodology described by Soderquist & Ealey (1994) and Soderquist (1995a) except that most location points of active individuals were obtained by sighting the animal and pacing on a compass bearing to a known reference point.

Logging Study

Investigation of the direct impact of logging on Brush-tailed Phascogales was undertaken utilising a BACI design (Before/ After/ Control/ Impact) (Green 1989). Prior to the logging study, two years of general data had been collected, with one years pre-logging data having been obtained from the Kingston sites. Logging operations occurred from March until mid July 1995 on one Kingston site (Figure 1), and nine impacted phascogales were radiotracked to their nests almost every day from the time of capture (February–May) until August 1 or death. Night tracking and observations of behaviour were also carried out on all of these individuals during this period. Other non-impacted phascogales were radiotracked in a similar manner, although radiotracking effort was much reduced. During the logging study a maximum of 30 phascogales were collared at any given time across all Kingston sites, and on the site affected by logging a maximum of 17 was achieved. Typically it would take a single person 6–12 hours to radiotrack 30 phascogales to their nest trees. All males in the population had died by mid July due to natural annual die-off, and the 1996 populations are currently being monitored to determine longer term effects.

Characteristics of Habitat Trees

Identified habitat trees were assessed for a variety of features, including species, dbhob (diameter at breast height over bark), presence of bark, condition (dead, senescent, alive), total height, height to canopy, canopy radius, height of fire damage, severity of fire damage, the estimated size of hollows available (large, medium, small) and for some trees, termite, rot damage and an estimate of the number of hollows. Hollow characteristics were measured in felled trees for 25 possum and 15 phascogale hollows identified in use at/before tree fall.

For the purpose of this paper, discussion will be confined to measures of tree species, diameter and tree condition. The diameter of trees is the best single predictor of the probability of hollows in Jarrah and Marri based on the examination of felled trees (McComb *et al.* 1994a), and therefore a central interest was to determine the diameters of trees which contained hollows suitable for phascogales. Fire is also an important issue in hollow development (Inions *et al.* 1989), and this will be dealt with separately.

Table 1. The frequency of tree species used for nesting by Brush-tailed Phascogales in Jarrah Forest of Western Australia.

Jarrah	Marri	undetermined *	Wandoo	Flooded Gum	Banksia	Total trees
284	362	12	13	17	1	689

* dead trees; species uncertain but known to be either Jarrah or Marri

RESULTS

HABITAT TREES

Tree Species Used as Habitat Trees

Six hundred and eighty nine individual trees were identified as habitat trees for phascogales. Of these, 664 (95.5%) were Jarrah or Marri with the remaining being Flooded Gum or Wandoo (Table 1). Both Wandoo and Flooded Gum appeared to have suitable hollows, but these species were very uncommon throughout the study sites, so low usage can be considered as a measure of availability rather than preference. Ground nesting in logs and under stumps was recorded but this was uncommon and confined almost exclusively to the mating period of one specific year.

Phascogales nested in Jarrah 41.2 % of occasions, and Marri 52.3% of occasions but underlying preference for individual tree species is unclear, because the relative proportion of suitable trees of different species was unquantified. Given that phascogales nested in Jarrahs of all sizes (Figure 3), no tree species preference was detected in reuse rates (see below) and there was a marked comparative decrease in Jarrah nesting incidence at above 80cm dbhob (Figure 3), the likely explanation for most of the variability in nesting between tree species is that of general availability. In particular, large Jarrahs were observed to be less common than large Marris over most of the area (all study sites had been lightly cut-over for Jarrah on two to three occasions) and Marris dominated the stream zones and gullies.

Size and Age of Habitat Trees

Tree size was recorded as dbhob for all trees. For Marri and Jarrah without bark, 4 cm was added to the measurement as bark thickness is relatively uniform at about 2 cm (P. Biggs, pers. comm., Dept. of Conservation & Land Management).

The 13 Wandoo recorded as habitat trees ranged in size from 39cm – 84cm dbhob, with a mean size of 60cm. Based on the diameter/age relationship established by Rose (1993), these sizes equate roughly to ages of 125 years (39cm), 175 years (60cm) and 300 years (84cm). No information has been found on ageing of Flooded Gum. Further reference to habitat trees in text and figures will be confined to Jarrah and Marri only.

Utilisation of nesting trees became most apparent at about 40cm dbhob (Figure 3) with a combined mean size of 82.8cm (standard deviation (S.D.) 28.7) The mean sizes of Marri and Jarrah used were 87.7cm and 76.6 cm respectively.

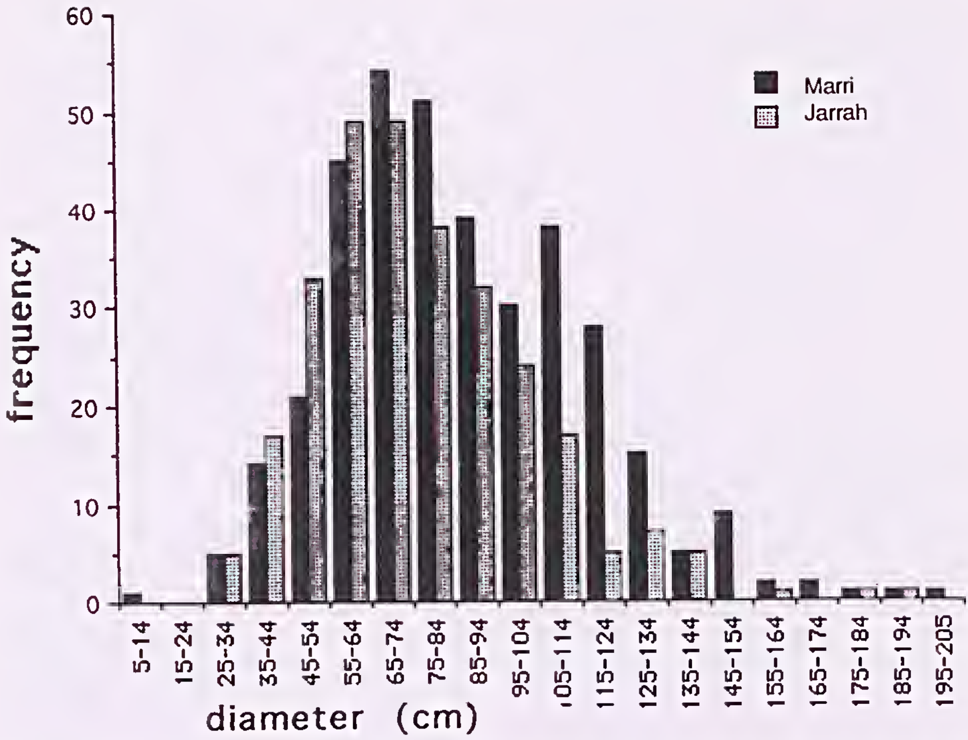

Figure 3. Number and diameter (diameter at breast height over bark) of individual Jarrah and Marri habitat trees identified in use by radiocollared Brush-tailed Phascogales (phascogales n = 139, trees n = 633).

Forty five percent of all trees used were less than 75cm, and 52.3% of these were Jarrah. Conversely only 37% of all trees above 75cm were Jarrah, but both these findings may have been a reflection of availability.

The estimated age of these tree species based on established diameter-age relationships varies considerably depending on which data are used for calculations. According to those presented in Inions *et al.* (1989), Jarrahs of the recorded mean size approximate 300 years, and Marris 580 years. However, preliminary examination of growth ring counts of felled trees near the study sites (unpubl. data, Dept. of Conservation and Land Management), suggests that age and diameter relationships between the two species are less disparate, and trees of the combined mean size of 82cm may be approximately 225 years.

Reuse Rate of Habitat Trees

Each phascogale utilised many habitat trees within its territory and regularly swapped between these trees, using some only once, while using others consecutively and/or repeatedly. The overall rate of detected reuse (use > once) of habitat trees was 60%. This does not mean that 40% of trees were not actually reused, but is rather a consequence of radiotracking effort and several factors which influence the use of any given tree: (i) an identified tree may not have remained constantly available to the collared phascogale because of the presence of other (undetected) phascogales (intraspecific competition/ avoidance) (ii) an initially identified tree may not have continuously fallen within the range of the collared phascogale – this was largely due to mobility of phascogales during dispersal, when their territories were unstable (iii) males and females swapped nest trees more regularly during the mating period, therefore the likelihood of detecting reuse was reduced at this phase and finally (iv) individuals may not have been tracked long enough or consistently enough to detect return to any given tree.

Therefore the absence of detected reuse of any individual tree does not necessarily mean it was of low suitability as a habitat tree. However, detected reuse does infer that such a tree is a particularly suitable habitat tree. After tree diameters were log-transformed to

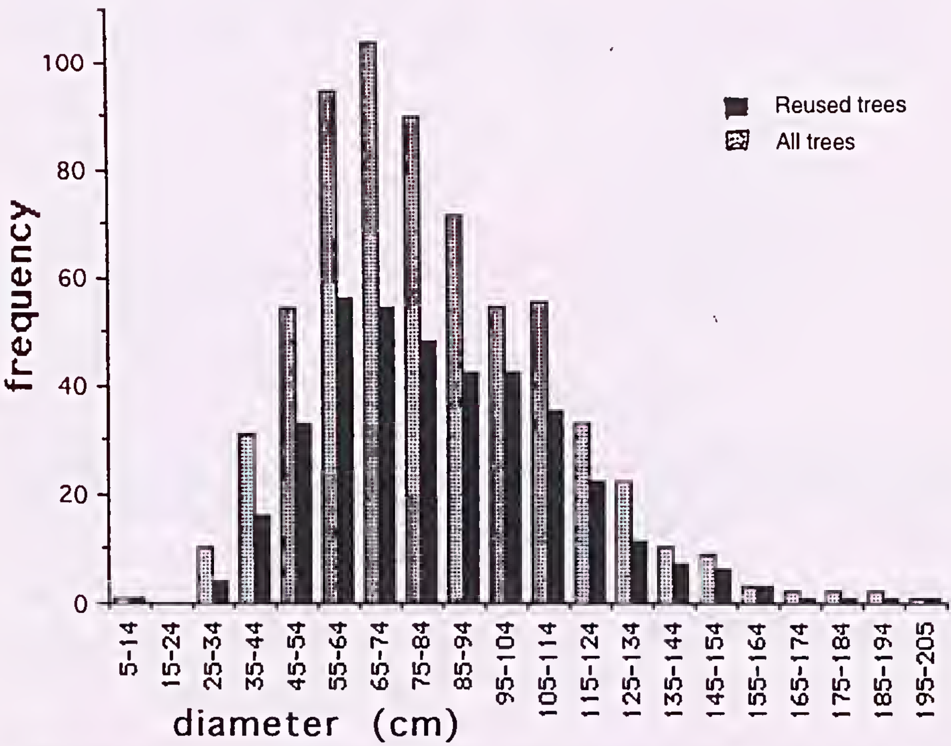

Figure 4. Number and diameter of re-used habitat trees compared to all habitat trees identified in use by radiocollared Brush-tailed Phascogales (phascogales n = 139, all trees n = 633, reused trees n = 384).

compensate for their skewed distribution (Figure 3) the mean diameters of trees used more than once were significantly greater than the means of those that were not reused (ANOVA, $F_{1,629}$ P=0.05) with a trend of greater reuse with increasing tree diameter (Figures 4 and 5). Reuse of habitat trees ≥ 95 dbhob appeared more frequent, but sample size effects became apparent for the very large trees which were

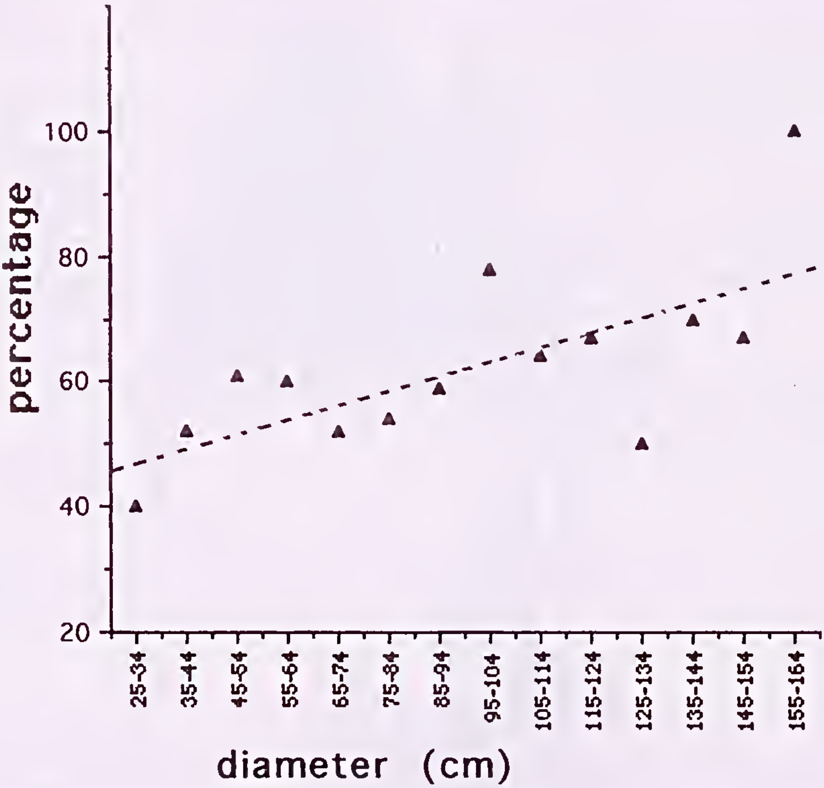

Figure 5. The percentage of habitat trees in each tree size class that were used more than once by Brush-tailed Phascogales (phascogales n = 139, reused trees n = 384, total trees n = 633) The line indicates direction of trend. Size classes with ≤ 2 trees have been excluded from the data.

Table 2. Mean diameters (dbhob) and standard errors (S.E.) of reused and not reused Jarrah and Marri habitat trees.

	Tree Species dbhob (cm)						
	Marri			Jarrah			
Usage	Mean dbh.	S.E.	n	Mean dbh.	S.E.	n	
not reused	84.30	2.5	136	75.35	2.4	113	
reused	89.79	2.0	218	77.47	1.95	166	(Total n = 633)

uncommon in the study area. The analysis revealed no significant tree species/reuse interaction. In all cases the mean size of reused trees was greater than the size of trees used only once (Table 2).

If reuse rates can be used to infer tree value (the greater the reuse, the greater the value of the tree as a habitat tree) then larger trees and especially those ≥ 95cm are better habitat trees than those of smaller sizes. This implies that identified mean habitat tree size is less informative than the rate of use of size classes when using such information to make management decisions. The sizes of trees used for nesting are a function of general environmental availability, whereas rate of use may detect preference and is more independent of availability.

Tree Condition

All trees were classified as dead, senescent or other (Figure 6). Twenty seven percent of recorded habitat trees were dead (dead trees were defined as 95% dead above ground – some had a few regrowth shoots), 40% were senescent (dying back) and the remaining 33% of trees were assessed to be in reasonable health.

Number and Spatial Organisation of Habitat Trees

Radiotracking of foraging individuals was undertaken to determine territory size based on activity, and habitat trees occurred throughout

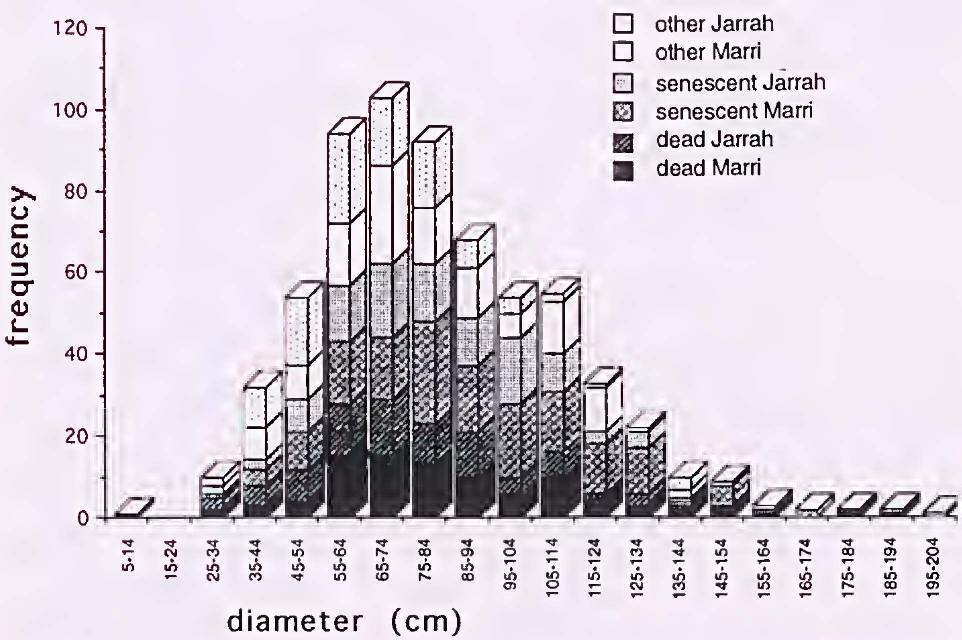

Figure 6. Number, diameter and condition of individual Jarrah and Marri habitat trees identified in use by radiocollared Brush-tailed Phascogales (phascogales n = 139, total trees n = 633).

each individuals territory without any apparent pattern (unpubl. data). The number of different habitat trees used by any individual varied greatly and both sex and temporal factors influenced tree use. To illustrate the extremes, one male used seven trees in 140 radio-tracking days and one female used 23 different trees in 37 days. The estimated requirement of 20 habitat trees/individual specified for Victorian phascogales by Soderquist (1995b) appears reasonable for Western Australian animals.

Phascogales that were associated with farms readily nested in trees in paddocks. Based on direct observations (and T. Soderquist pers. comm.), phascogales typically avoided moving over bare ground. However they would cross up to 300m of open space to nest in an isolated paddock tree. Further, the location of a nesting tree that was currently in use appeared frequently to be independent of where the animal was foraging. Phascogales were often observed to bypass several of their other closer nest sites when returning to the one in use.

Use of Phascogale Habitat Trees by Other Species and Hollow Entrance Requirements

Incidental observations of possums were made while observing phascogales on emergence from their hollows at dusk. Ringtail Possums rarely appeared to use dreys in this area (pers. ob., see Ellis & Jones 1992), and both Brushtail Possums and Ringtail Possums concurrently utilised hollows in dead or large phascogale habitat trees. This implies that selection of such trees (Figures 4 & 5) for retention in logging areas will likely encompass habitat trees suitable for both phascogales and possums. Additionally, because of the number and size range of hollows often present in these trees (McComb 1994), there is a reasonable likelihood that they may contain hollows suitable for other hollow-nesting fauna (e.g. birds and bats).

Table 3. Hollow entrance sizes used by Brush-tailed Phascogales and possums. Comparative data from other sources are included.

Species	Entrance size	Range (cm)	Mean	n	
Possums	width	4.5 – 17.0	8.8	23	
	length	7.0 – 27.0	13.3		
	diameter	6.0 – 49.0	17.6	32	(Inions *et al.* 1989)
Phascogales	width	2.4 – 6.2	3.9	15	
	length	3.0 – 21.0*	7.3		
	width	2.4 – 5.5		14	(Soderquist 1993a)
	radius	2.5 (semi circular shape in artificial hollows)		–	(Traill & Coates 1993)

In this study, large maximum entrance sizes occurred in phascogale hollows with * elliptical entrance shapes, and only one hollow entrance was recorded with both a large width and length (width 6.2cm, length 14.0cm).

Hollow competition between phascogales and possums is unlikely given the different hollow parameters and hollow entrance sizes used by each species. Based on the measurements of minimum and maximum entrance sizes (width and length) of hollows in felled or in accessible trees used by these species during this study (Table 3), possums utilised hollows with entrances larger than those used by phascogales.

THE DIRECT EFFECT OF LOGGING

Mortality at Tree Fall

The locations of radio-collared phascogales present in logging coupes were checked prior to each days' logging activities and only four trees containing collared animals were felled during operations. All phascogales survived tree fall without apparent injury. In contrast, the rate of felled trees containing possums was high. Based on one faller's records from the study site, over a 12 week period, 65 possums (10 Ringtail and 55 Brushtail Possums) were noted present in felled trees in coupes totalling approximately 63 hectares (1 possum felled/hectare). Eleven of these were killed outright (pers. comm. E. Vermey, Rose and Bending Forestry Service, Manjimup, Western Australia). This mortality rate of 17% only reflected death due to single tree fall. It did not take into account later death due to injury nor death as a consequence of being injured by machinery or other falling trees, or being in another tree felled later in the same day. These were actively avoided by the faller, but such actions would be atypical during most logging operations.

Habitat Tree Availability and Nesting Behaviour During and After Logging

To examine the effect of habitat tree removal on phascogales, nine individuals were radiotracked in areas affected by logging operations. Prior to logging, numerous habitat trees were identified as used inside the coupes that were to be cut, and hollow trees were considered abundant. The term coupe is used in this context to describe a discrete area that is cut; for logging operations in the Jarrah forest this is typically up to ten hectares of clearfell (gap) or larger areas of partial clearing (shelterwood). Regions of vegetation corridors of ≥ 100m widths are retained between each gap. For details on logging practises see CALM (1995).

The estimated amount of territory area affected by logging per individual ranged from 20% to 80%. Territory sizes for this species in continuous forest were estimated to be typically 20–30 hectares prior to the mating season (unpubl. data) with little difference in the size of female and male territories (c.f. Soderquist 1995a). During and after logging, behavioural observations of active individuals showed that all impacted phascogales continued to forage extensively throughout the cut coupes (both among ground tree debris and on the remaining trees), but ceased nesting inside the coupe boundaries. They confined their nesting entirely to the uncut areas retained between the coupes and there was no evidence of territory shift.

Seven coupes contained collared phascogales during logging. Of these, three coupes were logged without retention of habitat trees (as part of other researchers' study design). To examine habitat tree availability after logging, the four coupes with retained habitat trees were traversed completely (approximately 31 hectares) and examined for the presence of all trees marked for retention as potential habitat trees and all unmarked potential habitat trees remaining immediately after logging (Table 4). Identified trees were assessed and recorded to species, dbhob was measured, and they were estimated as either actual (1) or possible (2) habitat trees, and for marked trees, those that were unsuitable (3) were recorded as such. In brief, trees with possum tracks or numerous/ obvious hollow entrances (broken limbs, shed limbs, broken trunk, cracks in the trunk) were classified as actual habitat trees and suitable for possums and phascogales. Those with apparent hollow entrances and of either large size or with significant fire or termite damage were classed as possible habitat trees being marginally suitable for possums, and possibly suitable for phascogales.

Examination of each coupe revealed that few to no suitable habitat trees were retained. This is contrary to the logging prescription (CALM 1991) to mark and retain three potential habitat trees per hectare (or at a rate of 15 per 5 hectares). Either many of the trees marked as habitat trees for retention were unsuitable, or no such trees were marked. In all cases where trees were marked, the numbers were far fewer than are

Table 4. The number and estimated suitability of potential habitat trees marked for retention, and suitable unmarked potential habitat trees remaining in four coupes after logging.

			MARKED TREES					ADDITIONAL UNMARKED TREES WITH HOLLOWS	
coupe	estimated size hectares	logging style	Required H trees	Marked H trees	1	2	3	1	2
1	5	gap	15	0	0	0	*	0	1
2	6	gap	18	3	2	0	1	0	2
3	10	sw	30	9	0	3	6	1	7
4	15	sw	45	25	2	7	16	0	5
overall totals	31	*	108	37	4	10	23	1	15

H = potential habitat tree, sw = shelterwood, 1 = actual habitat tree, 2 = possible habitat trees, 3 =unsuitable as a habitat tree, required H trees = the number of marked potential habitat trees required to be retained during logging operations (CALM 1991) required to be retained. Retained trees ranged in diameter from 39 – 145cm (n=36), with a mean size of 76cm (S.D. 24).

Of the marked habitat trees (37 in 31 hectares) only four were estimated to be actual habitat trees, and ten had some possibility of containing suitable hollows. If one assumes that of the latter, 50% could be used for nesting by phascogales, the overall availability of habitat trees across the 31 hectares was 0.29 trees/hectare (1 tree/3 hectares). If the probability of use was less, the availability decreases further.

Predicting hollows from the ground always involves some subjectivity. To test method accuracy the data were examined using the logistic regression relationship developed by McComb *et al.* (1994a). Based on known hollow characteristics required by hollow-nesting species and the examination of hollows and tree characteristics in Jarrah and Marri (n=363), a relationship between tree diameter and hollow probabilities was established for various animals (not including phascogales). For Brush-tail Possums (model concordance = 86%,), where dbhob = diameter breast height over bark, the probability of possum hollows = 0.1*(1/1 = exp (4.14 – dbhob * 0.056)) (McComb *et al.* 1994b). Using this equation, collective hollow availability for possums based on the recorded diameters of 36 trees marked as habitat trees = 1.81 hollows/31 hectares. This compares reasonably with the assessment that four trees possessed hollows suitable for possums.

The contribution of incidentally left trees to habitat tree availability was low, and using the same principles as above, at best an overall (marked and unmarked) total of 0.53 potential habitat trees/hectare remained for phascogales (1 tree/2 hectares). In the shelterwood coupes where a greater retention of trees is intended, potential habitat tree levels for phascogales were collectively 0.45 trees/hectare (1 tree/2 hectares). In addition, unless incidentally left trees are marked for retention as "crop trees" they are not secure, as these are poisoned, felled or pushed over during post-logging Jarrah Stand Improvement (CALM 1995).

The absence of nesting by phascogales in cut coupes was directly due to few, if any, suitable phascogale habitat trees remaining after logging. Given that phascogales were observed to forage extensively throughout the coupes during and after cutting, spatial issues could not account for the change in nesting behaviour.

After logging, phascogale nesting was completely confined to trees within retained corridors, therefore these corridors of vegetation are of great importance. It is of concern that logging policy allows for the cutting of these corridors (excluding stream zones) in 10–20 years (CALM 1995). The additional removal of trees from the corridors in the manner noted above would result in local extinctions. Given the large territory sizes, exclusive nature of female territories and natural low densities of phascogales in continuous forest, stream zones alone will not provide sufficient area for populations to persist within logged areas. Although phascogale densities appeared markedly higher on the study sites than in populations in Victoria (females 0.05/hectare (WA) in contrast to 0.02/hectare (Victoria), Soderquist 1995a), this may be a phenomenon unique to the study area which is renowned for its high densities of several rare mammals (CALM, undated). For areas other than the Perup Forest and its surrounds, it is conservative to assume lower densities are typical and that disturbances would have greater impact on populations. Even ignoring the issue of loss of food resources caused by tree removal, other aspects of the species biology and behaviour compound their vulnerability to decline in areas of disturbance (Traill & Coates 1993, Soderquist 1994).

DISCUSSION AND CONCLUSION

The main purpose of this paper is to present preliminary findings pertinent to decisions which are currently being made on habitat tree requirements for mammals in the Jarrah forest, and to discuss recommendations and issues on habitat tree retention. The main findings of this research were consistent with those of comparable studies – the trees that contain suitable hollows are large, and logging removes these trees. Authors of research on hollow-dependent fauna consistently express great concern about the future of Australia's arboreal animals in production forests and reiterate that the reserve system has too many problems associated with it to mitigate concerns (see review McIlroy 1990, Lunney 1991, RAC 1992, Norton & Dovers 1994). What is encouraging is that modifications to hollow tree retention rates and logging intensity will alter the likelihood of hollow-nesting species persisting in logged regions (e.g. Kavanagh *et al.* 1995, Lindenmayer & Possingham 1995).

The mean size of habitat trees for phascogales in Western Australia was 82cm dbhob, which based on growth ring counts of felled trees equates to a tree age of about 225 years. More frequently used trees are larger, therefore much older. The implications of this are obvious; current documented practices combined with planning decisions based on short logging rotation rates are disastrous for hollow dependent fauna unless serious changes are made to the quantity and quality of trees retained inside logging coupes, the logging of previously retained corridors and buffers, and if need be, the size of coupes and logging blocks, and their proximity to other logging blocks and cleared farmland. The progressive loss of hollow-bearing trees across the Jarrah forest will eventually cause such a generalised refuge debt that hollow-dependent species will no longer be able to persist across their range. There is now sufficient basic biological and behavioural information available on many hollow-nesting species in Western Australia (Saunders *et. al.* 1982, Inions 1985, Wardell-Johnson 1986, Mawson & Long 1994, McComb 1994, Jones & Hillcox 1995) to assess some of their habitat tree requirements and to provide the species-needs basis for designing habitat tree retention policies in production areas of the forest.

Determining Habitat Tree Retention Rates in Jarrah Forest Affected by Logging

Designing optimum habitat tree retention policies for hollow-dependent fauna in timber production areas is complex, and much of the data needed to determine this are not available (see review of issues in Gibbons and Lindenmayer, in press). Most notably absent is information on the longevity of exposed retained trees in logged areas given varying coupe shape, size, and topography, the effects of soil disturbance during operations, and effects of regeneration burns and general controlled burns. Given these factors alone, retention rates must be conservative to allow for error in predicted tree longevity.

Ideal retention rates of hollow-bearing trees in areas to be affected by logging must take into account all individual arboreal species needs (the number and types of trees needed per hectare), the spatial arrangement of such trees, and the issues of tree survival and recruitment rates of hollow-bearing trees to replace others as they fall. For the Brush-tailed Phascogale, an estimate of four *actual* habitat trees per hectare is used given an estimated mature population density on the study sites of 0.2 phascogales/hectare (unpubl. data) and a habitat tree requirement of about 20 trees per phascogale (Soderquist 1995a).

Spatial Arrangement of Habitat Trees

The spatial arrangement of habitat trees may be important for some species (e.g. Ringtail Possum, Jones & Hillcox 1995). For mature phascogales, however, this is probably unimportant in gap clearings provided these do not exceed the currently prescribed size of ten hectares. Eight to 13 hectares is the estimated core territory size for female phascogales (Soderquist 1995), therefore gaps of $\leq$ 10 hectares would rarely exceed core territories, and would be $\leq$ 50% of total territory size. In contrast, as shelterwood clearing has no size limitation and there is no requirement for the retention of uncleared corridors between gaps and shelterwood, or within shelterwood, habitat trees within these areas would need to be spread throughout. Generally, habitat trees should be well spaced. Factors determining spatial arrangement of habitat trees should also be concerned with promoting the lifespan of the retained trees (Gibbons & Lindenmayer, in press) by configurations which offer wind protection, and by locations that do not allow waterlogging or drought stress, though never to the exclusion of choosing the most appropriate trees.

Tree Species

The species of trees chosen for retention may not be directly important, as suitable hollows occur in both Jarrah and Marri, and no preference was detected in reuse rates of these trees by phascogales. In the study by McComb *et al.* (1994a), tree species was not detected to influence hollow availability for Port Lincoln Parrots or Galahs, and was not considered an important factor in general hollow availability (McComb 1994). Inions *et al.* (1989) also concluded that possums did not select trees on the basis of species, but factors other than hollow availability (Gibbons and Lindenmayer, in press) suggest that a range of species should be retained in logged coupes.

Tree Size, Condition and Number Required for Retention

In contrast to tree species, the condition and size of trees retained as potential habitat trees is critical. With the exception of dead trees (see below) or trees with obvious hollows, the most reliable guide to hollow presence is tree size. Hollow size, range of sizes and number are directly correlated with tree diameter (Faunt 1992, Rose 1993, McComb 1994). Tree longevity, growth, the recruitment rates of hollow-bearing trees, the probabilities of any given size class containing suitable fauna hollows and the implications of these for determining potential habitat tree retention rates in logging regions of the Jarrah forest have been estimated by McComb (1994). In a modelled study using information about possum and some bird hollow requirements, he suggested the retention of three "actual" (>100cm dbhob or largest available) and three "future" (>50cm) habitat trees per hectare in high quality Jarrah forest (undefined, however is likely to refer to regions of highest rainfall and/or richer soils); in medium to low quality forest this figure is doubled (6 trees > 100cm, 6 > 50cm/hectare) as tree growth rates are slower and loss rates are higher in these areas. These specifications are predicted to provide one possum hollow per 3.4 hectares in high quality forest (1 hollow/1.7 ha in low quality forest) at the time of cutting and given loss and recruitment rates, one hollow per 3.8 hectares in high quality forest (1 hollow/ 4 ha low quality forest) in 100 years time. In addition, it is recommended that one clump of forest be retained per 10 hectares which contains: for high quality forest – 10 trees >100cm, 12 > 50cm and low quality forest: 27 trees > 100cm, 25 > 50cm. These clumps should be inspected to ensure that they contain these numbers – the implication being that minimum corridor size between coupes should be determined by the number of habitat trees present, rather than be of a prescribed size as is the current practice.

In addition to retaining live habitat trees, the retention of dead trees might be considered. Historically these trees have not been retained as habitat trees because of their financial value as firewood or woodchip, their hazard to logging crews during operations and their short lifespan as standing trees. They have also been shown to be the trees most likely effected by fire (Inions *et al.* 1989). However, dead trees have been demonstrated as being particularly important for many species including hollow-nesting microchiropteran bats (review Gibbons & Lindenmayer, in press), phascogales, Antechinus (Cockburn & Lazenby-Cohen 1992) and cockatoos (Saunders *et al.* 1982). As these trees do not represent a source of regrowth suppression, are im-portant for some species, and would provide hollow logs at fall, they should be additionally retained as current habitat trees and future habitat logs.

These recommendations are quite a departure from current practice. However, the policy prescription on habitat tree retention in logging areas of Jarrah forest is being revised and improved, with a requirement that four "primary" hollow-bearing trees > 70cm diameter be retained per hectare in gaps and shelterwood areas. The specified condition characteristics and spatial arrangements of these trees are appropriate. In addition to primary trees, six to eight "future" habitat trees/hectare of 30–70cm "if present, may be" retained in gaps (CALM 1995). There is no requirement for future habitat trees to be marked for retention in shelterwood areas. As yet the policy does not address the suggestion for variation between areas based on site quality nor changes to corridor sizes to include the recommended number of habitat trees between coupes. Additionally, the minimum size parameters of both the primary and future habitat trees are smaller than recommended. While in practice primary trees much greater than 70cm may well be left, it may be necessary to specify a larger minimum size as many trees around 70cm would not contain hollows with the deep cavities important for possums (Inions *et al.* 1989). In the case of future trees, the model predicts that if the minimum specified size (30cm) is chosen, they will not be available as suitable habitat trees before the loss of many primary trees.

Conclusion

Brush-tailed Phascogales in the Western Australian Jarrah forest require large numbers of hollow-bearing trees for nesting and examination of logging practices in this study showed that the availability of such trees was severely reduced within logging coupes. Changes to current habitat tree retention practices are needed if this species is to be conserved in production forests. Data collected during this study on the required number, characteristics and spatial arrangements of suitable habitat trees are consistent with recommendations made by McComb (1994) and short term habitat tree needs of phascogales may best be met by implementation of his recommendations. As these encompass the needs of several hollow-nesting species and include modelling of long term factors, they are currently the most appropriate available. However, data on the longevity of trees retained after logging is very limited and determining habitat tree survival should be considered a priority.

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